Section 4. Models For Ecosystem Restoration Planning

4.1 Corps Planning Needs and Useful Model Attributes

Many types of quantitative models have been developed to indicate ecological response (outputs) to natural and managed changes in ecosystem conditions (inputs). They vary widely in structure, assumptions, and ecosystem restoration planning utility. To be most useful for Corps planning purposes, ecosystem restoration models need to facilitate planning process that is consistent with Corps planning and ecosystem restoration policy. The most basic need is a model, or models, and methodological structure that organize ecosystem information so that it can be used to evaluate the effect of natural events and management measures (model input information) on ecosystem outputs indicative of both naturalness and associated significant resources.

Based on policies summarized in Sections 2 and 3 of this report, the most useful ecological models would be able to characterize the 1) existing degraded ecosystem condition, 2) the full range of more natural structure and function associated with partial to full restoration, 3) the *significant* ecosystem structure and function associated with partial to full restoration of naturalness, 4) the net changes in significant structure and function "in the planning area and the rest of the Nation" (from ER 1105-2-100), and 5) the sustainability of result over the long-term.

By Corps policy definition, the outputs representing environmental quality need to be ecological, meaning they should be the product of life processes, in total or in part. Therefore model development and choice should consider the influences of both the community and the habitat attributes of ecosystems, which interact to determine ecosystem output in its diverse expression. But, in addition, the more useful models will also consider controlling influences that arise proximally and remotely in the surrounding landscape, often well beyond the habitat-community complex in the project area. The influence of watershed conditions on lakes, rivers, wetlands, and coastal zones is the most usual generic example.

While the emphasis here is on *theoretical mathematical models*, ecological models do not necessarily have to be theoretical, mathematical or computer operated. Quantitative models may not be required for evaluation where and when existing *natural reference* conditions clearly provide a *physical model* that "maps" the desired outputs through restorative measures in closely connected but degraded areas. For restoration purposes, physical models are rarely small scale "mock-ups" of the real thing.   Most often the physical models are photographs, maps, and other representations of the desired *natural reference condition*. These can, in very specific conditions, clearly enough inform planners about the relationships between input measures and resulting ecosystem condition that there is no further need for mathematical models. Such clarity is typically rare, however, and good theoretical mathematical models add communication rigor, analytic flexibility, and model portability to the planning process in ways that typically elude physical models.

In addition to theoretical mathematical models, *statistical models*, which are *empirical* quantitative models, can be very useful in some circumstances, especially in situations where precision of prediction and uncertainty and risk analysis is very important and a sufficient history of relevant data is available. They develop measures of relationship between and among variables based on assumptions about theoretical models of variation and sampled-data distributions. They may be particularly useful in close conjunction with natural reference conditions, when it is possible to extend the specific conditions found in the natural reference condition into closely connected adjacent areas that have been altered.

Characterizing the more natural condition is only one aspect of modeling need. More natural structure and function of *recognized* social importance — the *significant* ecosystem resources— must be associated with naturalness to justify the investment. To be complete, ecosystem restoration planning models must identify at least two measures of ecosystem quality. One relates to satisfying the ecosystem restoration purpose, which is to restore ecosystem *naturalness.* The other relates to satisfying the need for a sound Federal investment, which is to restore ecological resources of recognized *significance*. These qualities may correlate closely in response to natural and managed influences on ecosystem performance, but often may not, as suggested in Figure 3.5 of Section 3. The *functions* supporting many natural services are likely to restore more quickly than the *structure,* which often includes the scarcest resources of greatest significance in an ecosystems biodiversity.

For greatest utility, ecosystem restoration planning model outputs need to capture both ecological resource quality and resource quantity. Corps policy indicates that the models need to characterize ecosystem quality and quantity through either a direct measure (physical units) or an indirect measure (indexes). Most restoration methods and some models are geographically based using maps of features that broadly determine habitat features and outputs. For Corps restoration projects, habitat dimensions are typically determined by water level in a channel or basin context of specified topography. Habitat area is determined, for example, by the boundaries of average water level in a river, by adjacent floodplain area in the channel, or by some fraction of wetland area within the floodplain. In coarse-grained models, the maps typically represent annual average, highs or lows, or other dimensions most relevant to the significant function and structure of inhabitant communities. Potentially useful methods recently developed track changes in habitat area through time based on the dynamics of hydrologic inputs, such as river discharge, in a topographic context.

Of course, geographic area and quality of habitat are related. The boundaries of habitat are determined where habitat qualities become so poor the space is uninhabitable. The dimensions and arrangements of different habitat attributes contributing to the environmental quality often vary with the geographic area included in a project. Boundary definition is clearest where the transition from habitable to inhabitable is sharpest, as it is at the water’s edge. Within habitable space, habitat is rarely of constant quality, either within or between habitat patches. Characterizations of relative quality have been much more difficult to address. Most habitat models focus on output indicators of habitat quality, the outputs of which are then coupled with acreage (or other geographical measure) determined from maps of plan-affected area based on some prescribed method/protocol. One of the most widely used of these methods is the Habitat Evaluation Procedure (HEP).

Model portability and "generalizability" are valuable attributes for Corps planning process. All models loose prediction accuracy as they are moved from one site to the next, however, if the new sites were not among those used to calibrate the original model. While a "one-size-fits-all" model is very process efficient if justifiable, the diversity of ecosystem and planning conditions thwarts such aspirations. Empirical models (physical and statistical) are especially limited in this regard because they are typically unique and applicable to the specific site of development. Theoretical mathematical models are typically more portable as a group, but also vary among model types.

Also unlike empirical models, theoretical mathematical models can be useful ways to organize new information incrementally based on lessons learned in each planning and implementation process and on experimental research. The better models, in this regard, act to integrate empirically established fragments of understanding by bridging remaining information gaps with field-testable possibilities. The most progressive management programmatically integrates empirical and theoretical approaches through a process of adaptive management (Walters 1986, Walters and Holling 1990). In this way uncertainty due to ignorance is gradually reduced.

Inherent uncertainty in forecasts will always remain, however, because of the importance of apparently random process in ecosystems. However, no commonly used management models have dealt with this issue much, let alone well. To some extent, uncertainty can be managed by increasing model scale and by choosing more integrative indicators of ecosystem output. Ecological effects of random events often exhibit consistent patterns even though specific distributions of effects vary widely and unpredictably. For example, the fraction and general pattern of wetland and upland areas in floodplains tends to be consistent even though the spatial distribution of wetlands and uplands may change remarkably following flood events. A small scale model that implies long-term sustainability of a specific wetland because it ignores the formative context of flood events flies in the face of geophysical and ecological reality. A large scale model that indicates the general pattern and fraction of wetlands and uplands in the entire floodplain controls for the uncertainty associated with specific distributions and is more likely to indicate the more important aspects of *resource sustainability.* When the models selected for use are small scale and the controlling dynamics are large scale, much more of planning responsibility rests on the methods used to properly interpret model outputs in the landscape context.

Also related to model scale, the most useful planning models would reveal the net changes in ecological resource output quality resulting at the National level as well as at the local planning level. This requirement for a National perspective in evaluating management effects broadens the spatial scale of planning perspective needed to determine the NER contribution to the nation. This broader perspective accounts for the degree to which the significant resources may simply be redistributing in the landscape without increasing total national output (analogous to the relationships of RED and NED). It addresses the possibility of ecological influences operating outside the planning perspective that could result in resource shifts within an ecosystem without any net increase in national ecosystem restoration benefit. In worst-case circumstances, resources could be shifted to a more risky habitat situation, resulting in a net loss of significant resource. For example, rare waterfowl identified as resources of significance might simply move from one migratory habitat to another, without significant gain in waterfowl numbers, but be exposed to greater hunting mortality. Such effects can operate at a small scale as well, especially in landscapes undergoing rapid changes, such as urban development.

Models with greater spatial inclusiveness also are more likely to reveal the ratios of local and national benefit to the investment costs. For example, if the desired resources are expected to double locally, but increase by a very small fraction of 1% nationally, the information provides insight into the relative local and national scarcity of the resource, which may be a consideration in justifying the Federal investment. The value of this small incremental gain is greatly dependent on *resource sustainability*, in this case indicated by local population *persistence*, which in turn depends on dynamics in the influential landscape. Examples might include restoring vernal pools for amphibians or fairy shrimp in privately owned watersheds that are rapidly becoming urbanized.

4.2 Attributes of Index Models and Actual Output Estimation Models

Quantitative models fall into two basically different output categories: *relative output estimation models* and *actual output estimation models*. Relative output models express model output as an "index" of the ecosystem output of interest —typically a habitat suitability index (HSI) for Corps projects. Actual output estimation models express model output in physical units that are intended to match the actual ecosystem output measured in the field. Examples of such output include water discharge per acre of restored wetland, numbers of juvenile birds raised to migratory staging per acre per year, or average plant biomass produced per acre per year. Planning policy allows either category of model to be used.

4.2.1 Relative output estimation models

Relative output estimation models typically take the form of species-habitat, community-habitat, biotic integrity, and functional capacity indexes. They define indexes of relative quality that are anchored in some optimal condition of maximum quality and varies downward toward zero as conditions change from optimum. Most "index" models useful for ecological assessment determine relative quality by some measure of species or biotic community output performance in a variety of habitat conditions varying from optimum to intolerable. For some indexes, the optimum condition is defined to be the most natural condition. For other indexes, the optimum condition does not necessarily have to be a more natural condition. The optimum habitat condition is defined by the maximum species or community output performance—usually some measure of abundance--which is assigned a maximum quality index value. The usual range of index values is 0 to 1, but any range can be specified.

Examples of biotic output measures include changes in population density of a species, population recruitment rate, species richness, functional capacity, and biotic integrity. Examples of relative measures of population density include bird calls heard per half hour and fish caught per 100 meters electro-fished. Species richness is estimated based on the number of species observed per unit of standard effort. Functional capacity is mathematically specified in a variety of ways, depending on function. One example, is the relative water storage capacity of an ecosystem compared to its most natural state. Biotic integrity is based on a suite of community performance indicators varying along a gradient from least human impact to most human impact. Conversion of measures to an index allows two or more different measures, including action estimates, to contribute to the calibration of an index, thereby making use of more information. Indexed qualities typically cost less to estimate than actual estimates. Being indexes, however, relative measures of biotic performance often incorporate unreported variation from sources other than the performance measure of interest.

Index models of species and community performance quality typically are structured independent of ecosystem area and need to be adjusted to make more meaningful comparisons among areas of different geographical size. This is done by normalizing geographical area to some standard unit of measure typically smaller than the area to managed, but large enough to incorporate most size related effects into the index of biotic performance. A commonly used unit is 1 acre. Quality indexes and geographical area are "integrated" by multiplying unit area (e.g., 1 acre) by the unit quality index and summing the multiples. One example of the product of this multiplication is the *habitat-unit* of HEP (FWS 1981), which in ideal circumstances can be compared directly to other habitat units of different spatial quantities and quality index values. This method relies on the assumption that a correction can be made through best professional judgment if there are important interactions remaining between the size and arrangements of geographical units and the quality of biotic performance. Where such interactions are common and intense, the utility of index models diminishes as more reliance is placed on professional judgment.

While they are usually less expensive to develop and apply than actual output estimators, relative output index models can incur unforeseen planning costs later in the planning process. As more nonlinear relationships and sharp inflections are incorporated in output indexes, the cumulative summation of "eco-units" becomes a less reliable index to total ecological output and complicates cost effectiveness and incremental cost analysis.
To be meaningful in tradeoff analysis, stakeholders need to be familiar with at least one condition along the gradient of relative quality so they can relate it to the projected change in index value. Stakeholders have an increasingly difficult time relating the change in indexed amount back to some reference condition that is meaningful to them. When these kinds of quality and quantity interactions are believed to be important, some form of ad hoc "adjustment" or "weighting" is required of the stakeholders, making the

model index meaning that much more difficult to interpret or to reproduce in similar conditions elsewhere.

A common field assumption that relative-output index models are more portable than other mathematical models can lead to erroneous conclusions about output amounts in plan evaluation and tradeoff analysis. It is often possible for an optimum index condition in one ecosystem site to produce much lower or much higher actual outputs in other situations. The index is most reliable for the conditions for which it was calibrated. Frequently, however, the calibration conditions for the original model form are not clear. As for any model, the need for calibration grows as conditions vary from the conditions for which the model was developed. Model calibration and verification ought to be based on the same performance indicators (e.g., bird calls per minute, fish caught per 1,000 m^2) used to construct the modeled relationships between input variables and output index. As much as possible this requires that the performance measure is taken under the same conditions for which the model was developed. Otherwise contextual variation (e.g., different seasonal and habitat conditions) can have important effects with misleading results.

4.2.2 Actual output estimation models

Actual output estimation models typically take the form of physical models, statistical models, and process simulation models. They generate model outputs that indicate actual ecosystem output amount or rate expressed in physical terms (e.g., discharge, biomass production, number of nests). Actual output estimations, make evaluation of model forecasts simple because real-world outputs can be compared directly to model output.

Physical models are small to full scale representations of the ecosystem state. While artificial models might be used to assess simple physical effects, such as vegetation effects on soil erosion (using artificial vegetation), most physical models are natural reference conditions of some kind. Small scale physical models are commonly used to evaluate ecosystem-level concepts, such as the response of vegetation plots to control of grazing, or the response of simulated rainfall runoff to vegetation cover. Such “pilot study” experimentation can be useful for testing ecosystem restoration techniques, such as plot response to restored elevations, substrate material and/or hydrology.

Full scale natural reference conditions often make excellent models for restoration, without any need for mathematical models. For example, a proposed restoration involves restoring downstream conditions to conditions like those upstream by 1) restoring the channel to a configuration like that upstream and, where possible, within the remaining outlines of natural channel in the project area (both sources of information are physical models), 2) by restoring diverted flow back to the channel (relying on the upstream condition of flow to indicate proper flow downstream, and 3) restoring a fish species of special status to the downstream habitat through natural colonization once upstream diversion impediment is eliminated (the presence of the fish species is a key part of the physical model). While transferring the model conditions to the project area may involve photographs, maps, and written specifications, no mathematical model is used in the process. While physical models have limited use under the described conditions, they are not discussed much further here.

Statistical models derive their structure inductively from samples of variables observed at specific locations. They summarize the behavior of variables in samples from the total universe of sample possibilities and estimate the range of variable behavior that might be expected among all possible samples taken. Statistical models do not identify cause and effect relationships. They simply describe the degree of relationship existing between or among variables. They often are used in combination with small-scale or full-scale physical models to characterize a mean value and variation in forecast output. Statistical models provide a measure of variation around the mean value, which can be expressed as a probability band within which the true mean lies.

Statistical models are used to test hypotheses and to extrapolate findings to a different time or location (forecasting). Hypothesis testing is used to determine whether one site condition differs from another site condition either in time or in space. Samples from a project site might be compared to samples from a reference site (the physical model of desired condition perhaps) to determine if the sites differ with respect to sampled parameters.

Statistical models are strongest for *prediction,* but only as long as the conditions they are calibrated for are clearly understood as cause and effect relationships and the context for restoration is very similar to the reference conditions characterized. As the ecological context changes, the prediction precision of statistical models tends to decrease rapidly to levels seen in other types of models, and they loose their prediction advantage. Statistical models are typically among the least portable but among the most useful when the precision of forecast result is desirable to know and to control. Because precision is a function of sampling intensity, their cost is a function of the precision desired. Process models also can include measures of confidence (or uncertainty) in the output estimate. A cruder sense of uncertainty can be determined for physical models as more natural references are visited.

Statistic models have provided much insight into the development of theoretical models and related research, but few have been used in ecosystem-level analysis. They have, however, been used to great advantage by the Corps and others for predicting river discharge based on long histories of discharge measurement at USGS monitored stations. Such databases rarely exist at the species and community levels of resource output from ecosystems. A large library of suitable references is available and they are not discussed much further here.

When there is no ecological interaction among habitat units as they are added, outputs estimates can be based on an "average" acre (other unit of geographical measure) of habitat or ecosystem output multiplied by the number of expected acres, such as 2 black ducks per acre of restored habitat for a total of 25 black ducks over 10.5 acres restored. Whether or not the areal dimensions and quality are the same for each unit, cumulative summation is relatively easy as long as each of the units are functionally independent, such as they might be for relatively small species in relatively large areal units of ecosystem. However, the ecological output per unit of ecosystem added often varies in practice as the quality of each added unit varies and sometimes it makes more sense to develop units of variable size.

Even if the ecosystem units are not independent, some spatially explicit process models are capable of capturing the quality changes that occur as units are integrated. Estimates of actual "physical-unit" output facilitate easy evaluation of cost effectiveness and incremental costs for different plans, and make tradeoff comparisons much clearer (e.g., 2.5 ducks/year verses $100 per year in water storage benefits). The primary disadvantage of these models is the difficulty often encountered in linking the specific outputs of interest back to fundamental indicators of production, biomass and numbers. Development and calibration costs often are relatively high. The main disadvantages of actual output estimation models are the primary advantages of the relative output index models.

Process simulation models provide many advantages. They have no inherently better predictive attributes than other models, and less so than statistical models, however. Because they are more explicit about process their workings are more transparent (to those who know the model language) than other models and they often make superior communication models among technical specialists. Unlike other types of models, they produce multiple outputs simultaneously and incorporate time-dependent feedback interactions that are hard to capture in index models and statistical models. This lends exceptional comprehensiveness and flexibility to their use. It is possible to link individual modules simulating the dynamics of resources of significance to a module designed to simulate a range of conditions along a gradient of relative naturalness. In this way the response of any number of resources of significance be generated simultaneous to the generation of measures of native biodiversity or other measures of naturalness (e.g., sustainability, resilience). Uncertainty due to random events can be built into the more sophisticated of such models (stochastic models). Some prototype process models are spatially explicit, providing outputs in mapped form.

Several weaknesses of index models are better addressed in models that estimate actual output amounts. Process models are especially useful in situations where many outputs are simultaneously of interest and time-dependent spatial interactions are important. This is usually the case in restoration proposals where many ecosystem alterations have occurred or are likely to occur and where a "shared vision" procedure tradeoff analysis is desired. Because they are superior models for organizing information into clear cause and effect pathways, and are particularly useful for sensitivity analysis, they are especially useful for adaptive management purposes. Process models show the greatest potential for generating integrated outputs of all NED and NER measures considered in multipurpose studies.

However, while many process models have been developed for research purposes, relatively few have been developed for management purposes. They usually require more time to assemble and/or to calibrate than index models, and tend to be more mathematically complex than other models. Their cost is typically higher than simple species habitat suitability models, but more comparable to community and ecosystem index models and to complex statistical models.

4.3 Important Models

4.3.1 Species-based Habitat Indices

Models with the longest history of Corps use are the single-species habitat suitability indices (HSI models), which were originally developed for mitigation analysis before there was a Corps ecosystem restoration purpose and NER objective. Unlike ecosystem restoration policy, compensatory mitigation policy does not require restoration of more natural conditions and habitats can be created to provide optimum conditions for species Single-species habitat suitability index values are maximum when an optimum condition exists for the species. The optimum condition for a species and the naturalness of the host ecosystem targeted for restoration may not coincide. Without knowledge of the relationship between the index value and the relative naturalness of the ecosystem, there is no way to confidently use such models to guide *restoration* to a more natural condition. Habitats can be *created* to desired levels of habitat optimality, however. Especially in situations where restoration is "simulated" through engineered means and natural conditions are not certain, single-species models are prone to guide development of a *created*, more *optimal* condition that is substantially different from a condition of greater naturalness.

HSI models are closely associated with development of the Habitat Evaluation Procedure (HEP) and, to a lesser extent, the Instream Flow Incremental Method (IFIM) developed under the lead of the U. S. Fish and Wildlife Service (FWS 1980, 1981; Bovee 1981). HEP typically was used for species in habitat settings other than flowing waters. IFIM was developed (Bovee 1981, Orth 1987, Nestler 1993) for aquatic species inhabiting flowing waters usually situated below water control structures where discharge is managed. More recently (Rubec et al. 1998 & 1999, Coyne and Christensen 1997), the National Marine Fishery Service has adapted habitat suitability measures to oceanic habitats.

HSI models were rapidly developed in the 1980s, in response to the need for evaluating compensatory mitigation determined under the National Environmental Policy Act (NEPA) of 1969. HSIs have been developed for many vertebrate species and some invertebrates. About 150 single-species HSI models are posted on US Geological Survey web pages and over 500 are believed to have been developed at one time or another. They vary greatly in quality, documentation, and the extent they have been verified and validated. Fewer models were developed for important ecological support species (mostly forage species) or for species indicative of certain ecosystem conditions.

The target of compensatory mitigation is very different from that of an ecosystem restoration target representing scarce resources in an unsustainable (degraded) state of degraded natural integrity. Firstly, compensatory mitigation did not require that a more natural condition be restored. More flexibility was allowed by accepting in mitigation the creation of new habitat optimum for selected species. The first species HSIs targeted relatively abundant species of high recreation and commercial value, and generally avoided rare species; especially those listed under the Endangered Species Act. Endangered species were excluded from compensatory mitigation consideration because they were too highly valued to risk their loss once listed under protection of the ESA. Negative impacts on endangered species were to be totally avoided in the first place. Similarly, if an entire ecosystem was very rare and composed of unique species, environmental impact analysis and mitigation would usually choose avoidance of negative impact over attempted compensatory mitigation. Compensatory mitigation was most often allowed by the regulatory agencies when the losses were economic (recreational, commercial fishing) rather than environmental (EQ).

The HSIs, HEP and IFIM generally worked well conceptually for "exact" compensation of fish or wildlife loss as long as the same measures were used to assess both the impact site, before it was impacted, and the compensatory habitat created or restored for mitigation. Loss of a large acreage with low average quality could be compensated by creating or restoring a small acreage with high average quality. The assumption was that, regardless of quality and quantity combinations, the *value* of habitat lost to water resource development was at least fully compensated by the *value* of restored or created compensatory habitat.

An important complication occurred when the consistent use of the same species index over impacted sites and compensation sites was impractical because the value of HUs varied among different species. Two or more species with the same HSI, or increment of change in HSI, usually differ widely in abundance, production, or other measures proportional to species value. In addition, human preferences for different species often vary depending on perceived utility and/or value. Even for endangered species, "charismatic megafauna" (e.g., bald eagle, salmon) are valued more highly than small and cryptic forms (e.g., freshwater mussels, snail darters). There appears to be no practical way that habitat units of different species can be made reliable indicators of relative value for comparative analysis.

Another problem arose when ecological settings for the compensation site and the impact site differed substantially. The interactions among habitat variables then became different and increased the probability that the same index represented different species abundance or other performance measure. For this reason on-site compensation was preferable to off-site compensation except when it was impractical. In addition, the farther off site the compensation occurred, the more it altered the supply of resource with respect to human demand. The same resource production could become less or more valuable as a consequence. (This is not a problem for species recognized nationally as important because of their vulnerability to extinction but having no overt utility, because no local interest has the advantage of proximity.)

As used in the past, single-species HEP often did not capture all of national interest associated with ecosystem services (NRC 1999). This problem is associated with the degree that the habitat of a single species indicated *all of the value* that needed compensation. For this reason many of the species selected for HSI development were dominant species of high recreational and/or commercial value. They captured much of the value in their index. In some cases, the species were selected as habitat indicators for a collective productivity of valued resources, such as an abundant forage species that sustains a number of more directly valued sport and commercial species. Even so, it was difficult to assure that all ecosystem services and values associated with an impacted site were captured in the habitat requirements of a single species. As a consequence, the habitat focus of compensation typically ignored effects on water storage, water treatment, storm-surge reduction, and other ecosystem services that could have been important.

Finally, there is the issue of predictive accuracy. Brooks (1997) has criticized insufficient verification for existing HSIs and there is some evidence that existing models have not proved as effective as once hoped. The more universal chronic complaint is about the lack of evidence for or against the continued use of an existing model. However, this general complain also applies to other ecosystem management models used by government agencies.

Many of these issues have proved to be problems for ecosystem restoration use. Another issue is the degree that a single species can inclusively indicate more natural conditions for the entire habitat and community complex comprising the ecosystem. The most influential attributes of a species' environment form a subset of all attributes affecting the community-habitat complex. The best indicator species are often dominant plants, for which few HSIs have been developed. However, community-habitat indices may be a better general alternative to indicator species for representing the relative naturalness of ecosystems.

Even so, the most socially *significant* resources of ecosystems are likely to be scarce species in many decision processes. HUs based on the needs of the scarce species could be useful, once developed, but few now exist. Because they are too narrowly focused to be inclusive indicators of a more natural *ecosystem* condition, the most effective planning use can be made of them when they are linked with a community-habitat model of relative naturalness and integrity. In that process, the degree of restoration applied to a more natural ecosystem condition can be evaluated against incremental cost and outputs from that model can serve as inputs to the single-species habitat models to evaluate the effect on the significant resource.

4.3.2 Community-based Habitat Indexes

Community habitat suitability models offer improvements over species level models for indicating the naturalness of ecosystems in a number of ways. The WCHE, RCHARC and IBI models, for example, are based in structural indicators of community naturalness. They anchor their maximum index value to a native species diversity or other native biodiversity measure existing in the most natural state determined from reference conditions. Because they indicate the relative effect of humans on communities they can be useful for formulating and evaluating according to the ecosystem restoration planning objective. However, unless the entire community is an ecological resource of national significance, or there is a known direct relationship between the community index and the condition of the significant resource, some other measure of resource significance is needed to evaluate the restoration of resources of significance.

An important limitation of community-habitat models of all kinds is that they are typically based on the habitat needs of the more common species in the ecosystem—that is, the species that are most readily investigated for model development. Figure 4.1 illustrates different possible relationships that might exist between the habitat suitability indices of vulnerable resource species, which probably qualify as resources of national significance, and a native species richness indicator generated by community-habitat model. If the vulnerable species and the native species follow the same patterns of relative abundance and rarity in the system, a relationship like B in Figure 4.1 would exist and could be used to guide restoration species viability as well as the full complement of species and functions. Even so, extrapolations of relationships to the rarest species is prone to uncertain results. In situations where the vulnerable species are very rare in the ecosystem and are likely (example C) to be restored to the community-habitat complex only as the ecosystem approaches a fully natural state, most of the vulnerable species will fall into an uncertain restoration status.

The broadly adapted species in ecosystems are often among the more common species that dominate the restoration of a disproportionate amount of the function other than that related to sustaining the most sensitive species. In relatively few ecosystems (some isolated western spring systems, for example), globally rare species dominate (example A). These are typically unique ecosystems, however, for which there are no generally applicable models. Development of a biodiversity model of relative naturalness for such conditions would include most of the globally rare species because they often dominate in these simple systems. In most ecosystems, however, there is little evidence that globally vulnerable species are consistently among the dominant species.

The relevance of the relationship that actually exists in restoration prospects is important for prioritizing restoration actions. If the primary justification for the proposed restoration is reducing species vulnerability and model B is correct, then a partial restoration action would contributes proportionally little to that end. Most or full restoration should be the objective to assure a significant fraction of the vulnerable species will recover. If the justification is based more on the recovery of services other than the genetic information in rare species, partial restoration may be more suitable. For example, restoration of erosion control and nutrient retention may occur relatively quickly as biomass accumulates in the restoration of a small fraction of the community. The stability of function is likely to increase with further restoration of community components, but not at a consistent rate like that indicated in model B.

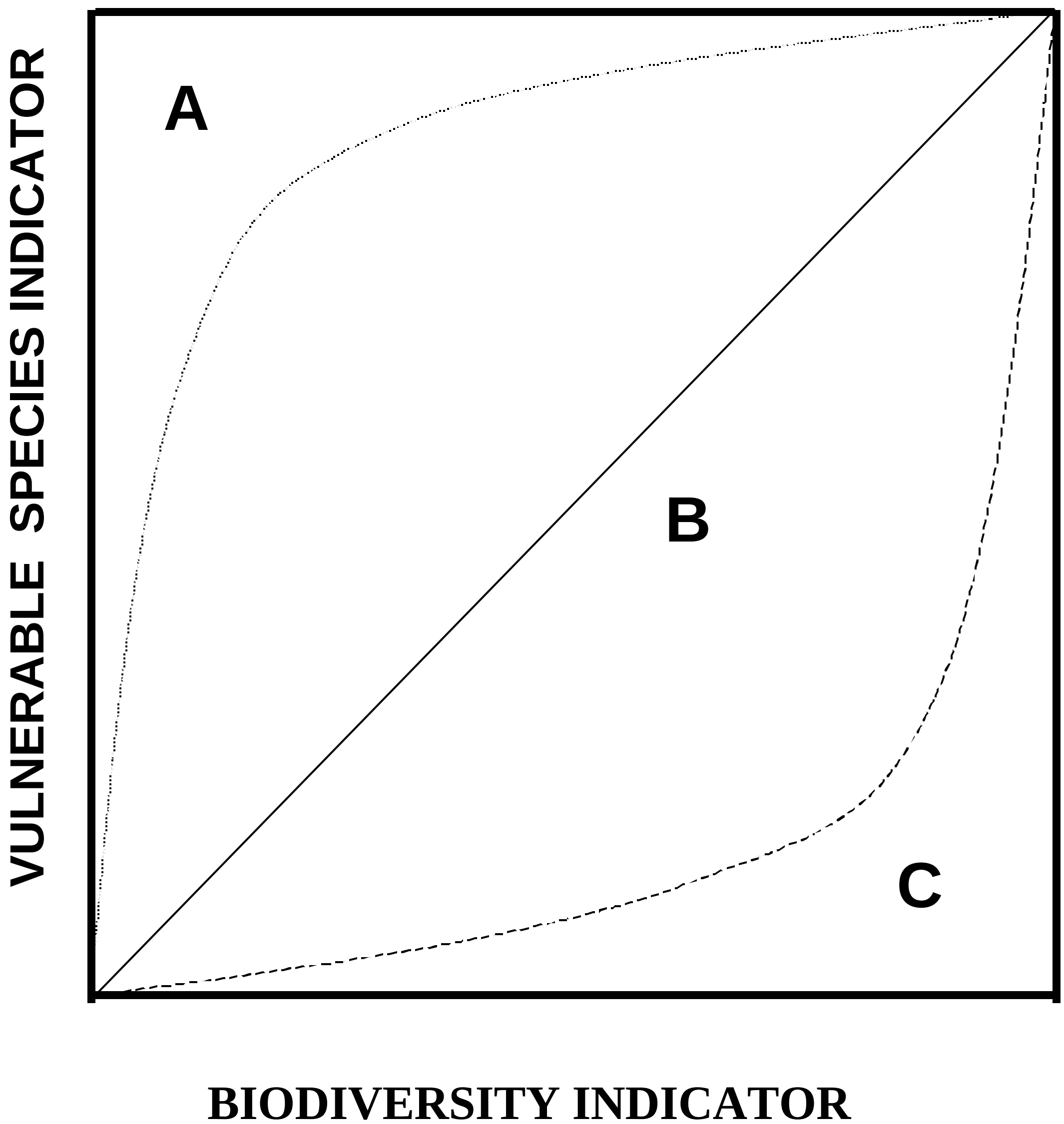

Figure 4.1. Possible relationships between a theoretically inclusive habitat suitability indices for native species richness and for the vulnerable species in ecosystems, depending on whether the vulnerable species are more abundant (A), less abundant (C) or equally distributed in abundance (B). In most conditions, only a fraction of the more common species are included in the development and calibration of such models (as indicated by D) because of sampling limitations. See the text for discussion of the Relationships.

The same caveats hold true for the HGM approach, which assesses the naturalness of ecosystem function based on a suite of functional capacity indexes calibrated against a gradient of human effect, with the most natural state of a carefully classified wetland type having the maximum index value. But because functional capacity often recovers quickly with the restoration of the more common species (as discussed in Section 3) scarce resources may not be recovered in anything short of full restoration. Even then, this and other ecosystem-level models tend to overlook important connections to the larger landscape.

Similar to single-species habitat indexes. community-habitat or ecosystem index models are typically limited to a local planning perspective. They tend to externalize large-scale landscape features and processes that can be very important for assuring the natural processes of importance to significant resources are restored. RCHARC, for example, indicates variation from a natural state based on the velocity and depth alterations, but does not address other factors that might impede recolonization off site, once desired conditions are created or restored. This requires an alternative, usually ad-hoc (professional judgment), evaluation of landscape level influences.

None of these index models forecast or evaluate the change in the national resource condition, which would be challenging, but theoretically possible, The index might include weights proportional to local and national contribution to the resources of significance, such as relative abundance or geographical area. This type of information is invaluable for evaluating the significance of plan effects. It is complicated however by the fact that indexes nay not reflect differences in habitat-community productivity very well. Some do not have any production factor and others simply average production factors (relative abundance measures) in with other factors. The optimum conditions determined for resources of significance in two different ecosystem areas might produce an order of magnitude difference in the production of individual organisms contributing to the national resource. These differences may or may not be integrated into the calibration of the models. Existing indexes need to be reconsidered in a national perspective to assure that they account for such differences.

When the resources of significance are single species, a single species-habitat model, or an array of such models, can be checked against the proposed state of naturalness indicated by a community-habitat model to determine how the relative performance of the resource is likely to respond to the more natural condition with respect to its optimum. Like other models applied only in a local planning context, if there is not a clear idea increased abundance, reproduction rate, or other measure of resource improvement, there will be no clear idea of how that improvement relates to the state of the resource nationally.

The predictive capability of both natural integrity and specific resource response to restoration measures decreases sharply as ecosystems become more generally modified and undergo more intense stress. Existing models are based on the assumption that the traditional concept of resilience is in effect and that processes of natural colonization and succession are consistent with that concept. As the probability that ecosystem response will "flip" into a new stability regime increases because of widespread disturbance, existing models become less reliable for formulation and evaluation. Two strategies for controlling this source of risk include 1) emphasizing restoration where the traditional "rules" of resilience are most likely (the short term strategy which avoids widely disturbed areas of ecosystems), and 2) developing more spatially explicit and comprehensive models based on improved understanding of culturally fragmented and stressed ecosystems.

Brief descriptions follow for nationally recognized community-habitat index and ecosystem functional capacity index models:

Wetland Valuation Assessment (WVA).. This is an interagency product developed for use in coastal wetlands, mostly in Louisiana, to carry out authorities under the Coastal Wetlands Planning, Protection, and Restoration Act of 1990 (Louisiana Coastal Wetlands Conservation and Restoration Task Force 1991, Environmental Work Group 1998). It is a community-level HSI/HEP approach. The wetland types include freshwater marsh, brackish marsh, saline marsh and cypress-tupelo swamp. Just as for species HEP, expediency in carrying out the federal law was an important criterion for assembling the community HSI models. The WVA is the only community model described here that does not establish its maximum index value based on some undisturbed natural state. The maximum habitat suitability is based in a concept of some community-level "optimum" based on an "average" optimum condition determined from the HSIs of 31 high-profile indicator species. It is, therefore, not an ecosystem restoration model in the narrowly defined sense.

Index of Biotic Integrity (IBI) . The Clean Water Act set as its objective, the restoration of physical, chemical and biological integrity of the Nation's waters—i.e., restoration of aquatic ecosystem integrity. Recognizing that biological integrity depends on suitable physical and chemical conditions, Karr (1981, 1986) devised an Index of Biotic Integrity (IBI) to assess progress in meeting an ecosystem restoration objective, such as might follow from the elimination of a chemical pollutant. The IBI is a multidimensional index of different community habitat conditions summed from a suite of subordinate indexes based on the richness, composition, and health of representative members of a community group (Karr 1991). It has been developed for fish and invertebrates. Of the indexes described here, the IBI is the only one described among ecological indicators for the Nation by the NRC (2000).

The IBI is based on the regional native biota indicative of unique communities and is anchored in the community and habitat integrity of undisturbed ecosystems. It has been most thoroughly developed for Midwestern streams, but is undergoing development and evaluation in wetlands (Minns et al 1994, Burton et al. 1999) and other stream ecosystems (e.g., Simon 1999). The Midwestern fish IBI is composed of 12 subordinate indexes, each of which is ranked 1, 3 or 5 indicating the variance of a community from the unimpaired natural community condition. The best score is 60 points and the lowest is 12 points. The IBI has stimulated widespread interest in applications elsewhere in recent years. Plafkin et al. (1989) described rapid assessment protocols using an IBI approach and discussed the potential for guiding restoration. Because it is designed specifically to restore more natural ecosystem integrity, the IBI leads among models for guiding the restoration of more natural ecosystem conditions.

Wildlife Community Habitat Evaluation (WCHE). The Clean Water Act authorized the Corps to regulate discharge of dredge and fill material into the Nation's waters with the intent of mitigating impacts where practicable. Wetlands have received exceptional attention due to state interest and U. S. executive-branch policy. Numerous wetland evaluation methods had been developed (Bartoldus 1997) but none satisfied Corps regulatory needs. Schroeder and Haire (1993) had reviewed existing community-level habitat indices in response to a need expressed by the U. S. Fish and Wildlife Service for practical assessment tools more comprehensive in scope than the single- or multi-species HSI models existing at the time (e.g., FWS 1981, Short 1984, Adamus 1987). Out of that philosophy Schroeder developed a small series of upland models and won the attention of the Corps who funded development of a WCHE for forested wetlands in Maryland (Schroeder 1996a and b).

The forested wetland WCHE developed community-habitat suitability indices for community assemblages of native species based on the relationship of native vertebrate species richness to several habitat variables including habitat edge and isolation attributes (Schroeder (1996a and b). The native vertebrate species richness is the criterion used to gage community response to habitat suitability . A maximum suitability is indicated for the condition that supports the maximum number of forest-interior native species. An important conceptual advance was incorporation of landscape-level habitat features that reflect the effect of habitat fragmentation. However, a disadvantage in the single wetland model so far developed is the lack of hydroregime habitat variables that might link vegetation form and other ecosystem attributes to Corps restoration measures. Because it is based on a scale of relative naturalness, this model has potential for utility in place of or in addition to the IBI and other community-habitat index models.

Riverine Community Habitat Assessment and Restoration Concept (RCHARC). The Corps also has invested in the development of a model for use in environmental mitigation of physical impacts on flow regimes in large rivers and for guiding river-ecosystem restoration decisions (Nesler et al. 1995). RCHARC derives its underlying concept from single-species HSIs developed for IFIM . It relies on the relationship between most fish species contributing to the membership of the river community and the distribution of flow velocities. Unlike the WCHE for forested wetlands, RCHARC is linked to hydro-regime management.

RCHARC was developed and used for the Missouri River and has had limited application elsewhere (e.g., Apalachicola system). Like other habitat-based relative indices of community condition, the maximum index value is anchored in that habitat condition resulting in maximum species richness observed in a range of flow conditions. Being narrowly defined in terms of flow dynamics, RCHARC as it is presently configured predicts habitat suitability only for flow dynamics. The model cannot predict accurately for a site where other variables are limiting, such as oxygen or temperature. Like other community-habitat models that attempt to characterize a range of relative naturalness in ecosystem condition, this model is suitable for restoration purposes, but would be more suitable if other habitat variables were included in addition to hydrologic variation. Also, in the highly modified, large river conditions for which it was developed, it is difficult to separate natural variation from variation caused by human impacts.

An Ecosystem Functional Capacity Index—The Hydrogeomorphic Approach. Only one method develops indexes of ecosystem function. Following an executive order for no-net-loss of wetland function and value in 1990, a technique was sought to assess wetland ecosystem functional capacity. The Hydrogeomorphic (HGM) Approach was developed in prototype by Brinson (1993) and its development is continuing. With Corps funding, Smith et al. (1995) expanded the concept and initiated development of specific models for different wetland types. The basic premise made for calibrating HGM models is that unimpaired ecosystems within each ecosystem type are fully functional (1.0) and human alteration reduces the functional capacity index (FCI) along a scale between 1.0 and 0. A wetland classification has been completed to determine the fully functional benchmark ecosystems and a number of type models have been completed. Wetland types are defined by hydrologic, climatologic and geomorphologic settings and associated communities (Brinson 1993). While theoretically applicable in any ecosystem type, the method has so far been applied only to wetlands.

Somewhat like the IBI concept, the HGM Approach uses a number of functional capacity indices to define the ecosystem condition. These vary in type and algorithm depending on the wetland type. Unlike the IBI, however, the FCIs were not intended to be summed, averaged, or otherwise integrated into a single index value. Wetland functional attributes depend on wetland type and cannot be compared directly across wetland types. While some types of functions are held in common among all wetland types, such as water storage and habitat functions, many functions are limited to a subset of wetland types. Organic detritus export, for example, is a function only of wetlands occupying open basins. Each function is described by its own functional capacity index, which is calculated by an equation assembled from a number of indicative community habitat variables (e.g., suspended solids and water level fluctuation).

The HGM Approach has potential for use in guiding wetland and other ecosystem restoration actions. It has one important advantage over the community HSI models in that it is more inclusive of all ecosystem functions relevant to ecosystem services. King et al. (2000) are studying the possibilities for a weighting method to create a wetland value index from functional capacity indices based on ecological context, social context and human preferences. The HGM Approach, however, retains most of the shortcomings of any relative index model. The predicted results have little meaning outside the ecosystem reference framework. Different ecosystems can only be compared through the functions they hold in common. In addition, the indices to the different functions do not directly reflect the biodiversity variables that appear to influence functional stability in support of service reliability. Even so, the HGM Approach characterizes the relative naturalness of ecosystems through their important functions and can be useful for evaluating measures taken to achieve the ecosystem restoration planning objective.

4.3.3 Ecosystem Process Simulation Models

Models that simulate ecosystem function and structure are based in concepts dating back to Lindeman (1942) and Odum (1957). They are variously known as process models, simulation models, compartment models, input-output models, mechanistic models, modular models, and dynamic state models. Their common intent, however, is to simulate natural process rates and output amounts as closely as needed for the model purpose. They are typically developed from theoretical mathematical descriptors of process and form but may be hybrid models including both theoretical and empirical elements (statistical equations). Many such models have been developed for research purposes, such as formulating a hypothesis of how complex ecological mechanisms might interact to generate an ecosystem output, which is then compared to real-world observations. Fewer process simulation models have been developed and widely used for management applications because they usually take more time to develop than allowed by statute mandates. They frequently require local calibration with extensive date, are relatively costly to use, and often involve a disconcerting array of variables and outputs for practitioners typically focused on one or two model outputs. NRC (2000) refers to a number of qualitative concept models and related quantitative models of ecological process relevant to development of national ecological indicators.

The multitude of possible outputs and comparisons also can be advantageous for analysis of complex ecosystem process. Unlike index models, process simulation models can provide great flexibility in use and can enable direct comparison of numerous interactive outputs in response to inputs of simulated environmental stress or management change. Among the more useful capabilities is for analysis of management tradeoffs among ecological outputs in a “shared vision” approach to planning. In addition, the outputs from one model can be coupled to the inputs of other models in time steps that allow simulation of natural feedback effects and interactions among different modeled functions and structures.

Community-level structural and functional output from one component (e.g., vegetation form and production) can provide controlling inputs to species groups and to individual population components. Any number of significant output modules can be modeled at the species or ecological guild level. It is conceptually possible, therefore, to include both ecosystem-level measures of naturalness in model form and function and subsystem models representing resources of significance, and even feedback interactions between the two if appropriate. Hybrids of species-habitat index models and process simulation models have also been constructed (DeAngelis et al.1998), but feedbacks from the index models are conceptually difficult.

At the model core are state-variable equations that quantify a condition at a particular time, but vary through time as model inputs vary. A common state variable condition is biomass density (e.g., kg/hectare) of a functional community group, such as primary producers or herbivore secondary producers. The state variables change as input conditions change with each time step included in the model. Time steps vary greatly, from minutes to years depending on the scale of interest and data availability. The state variables form compartments with driving inputs and outputs that serve as inputs for other compartments. The state variables are linked by equations defining relationships with coefficients influenced by other variables. Density-dependent feedback relationships are common in ecosystems and in process models. The amount of change in a state variable often determines in part the amount an influential variable changes. Food-web feed backs combine with habitat variables to determine the functional stability of state variables.

The basic input variables used in aquatic ecosystem simulation models typically include initial biomass of producer groups, the driving energy input (usually solar and biochemical), controlling nutrient concentrations, water flow, topography (channel and basin form), temperature and other environmental-constraint data. Temporal variation in solar energy and water discharge are necessary inputs for fully simulating ecosystem dynamics. Depending on purposes, simulation models may either explicitly or implicitly cycle nutrients and track other material flows.

Spatially Constant Models. An early example of an aquatic process model is Clean-X developed for open waters of lakes (Scavia et al. 1974) and a stream model by McIntire and Colby (1978). The most important conceptual model for streams, the River Continuum Concept A more recent aquatic ecosystem simulation model, CASM (Comprehensive Aquatic System Model), has been used to assess ecosystem structural and functional relationships (DeAngelis et al. 1989) and risks of dysfunction from contaminants and other stressors (Bartell et al. 1999). Friend et al. (1997) described a process-based, terrestrial biosphere model of ecosystem dynamics (Hybrid v3.0) for global assessment. This is a general application model of carbon, water and nutrient cycles coupled with soil, plant and atmospheric systems. Models of this scale may have potential for analyzing cumulative effect of restoration process to regional or global process. The Corps has invested in a Successional Dynamics Simulation (SDS) model for upland terrestrial conditions affected by military operations (McLendon et al 1998).

Spatially Variable Models. Spatially explicit process models are relatively recent additions to simulation model advances. Their development has been closely coupled with Geographical Information Systems (GIS). Especially targeted for modeling attention have been the movements of living organisms through landforms and across landform boundaries (the so called Mobile Animal Models [MAP] described by Dunning et al 1995. Rudimentary spatially explicit community models have been developed, such as the wetland model described by Poiani and Johnson (1993). One of the more elaborate examples of spatially explicit models is ATLSS (Across Trophic Level System Simulatino), which has been developed for South Florida study of Everglades restoration (DeAngelis et al. 1998)

Recently two spatially explicit models have been developed with potential for aiding restoration process: FRAGSTATS (McGargigal and Marks 1995) and PATCH (Schumacker 1998). FRAGSTATS provides the user access to a number of algorithms for calculating landscape-scale metrics such as habitat area, patch sizes, patch pattern, and total edge development. FRAGSTATS has been used to assess landscape suitability for both single species and groupings of wildlife (Rosberry and Sudkamp (1998), Glennon and Porter (1999), Penhollow and Stauffer (2000). PATCH provides a GIS-based platform for tracking wildlife populations through time and space While PATCH will track several populations in a landscape context simultaneously, it does not account for population interactions. FRAGSTATS and PATCH offer an advantage over HSI in their potential for evaluating the importance of habitat connectedness to other habitats for restored habitat colonization from dispersing populations. To the extent they are most useful mostly for simulating environments of individual populations, they have some of the same limitations as species-based habitat suitability indices.

A Geographical Information System (GIS) is the usual means for organizing and overlaying data in a map-like or geographical format. GIS is not a model, but a database management system that is increasingly integrated with ecological models. A GIS may be used to input information, house model process, and output information in map form. GIS software has greatly facilitated model use and development for spatially explicit natural resource inventory and management. A common use of GIS is to store ecological data on land form, vegetation, land use, species and other distributions according to map coordinates. A national-scale example of this use is the development of GAP Analysis (Scott et al. 1993) by the U.S.G.S. For GAP analysis, vegetation, species distributions and property ownership boundaries are overlain to assess the species distributions with the intent of identifying key areas of high biodiversity and high vulnerability based on potential land and water use. All of the United States is expected to be completed over the next few years. GIS also is widely used to organize information at much smaller geographical scales. The upper Mississippi Corps districts, for example, use it to carry out the Upper Mississippi Environmental Management Program and interfaces it with a simple process simulation model that predicts plant succession to forecast habitat condition changes. A good example of GIS use in a process simulation model is ATLSS, which is used to analyze plans for restoration of the Everglades and adjacent ecosystem conditions in South Florida (DeAngelis et al.1998).

4.4 Choosing Models for Restoration Planning

4.4.1 Importance of the Systems Context

Determining the "best models" to use for guiding restoration of more natural ecosystem conditions and associated resources of significance is situational, depending on the complexity of the natural state and the alterations that have occurred. Just about any rigorously applied model type, including physical models, may suffice for situations where there has been very little ecosystem change from the natural state, the condition to be restored is closely connected to the restoration site, full restoration is feasible (at least to the level indicated by an existing natural reference condition), and the source of the deficiency in resources of significance is easily identified and removed. However, most models do not explicitly evaluate sustainability, but rather assume that a close relationship exists between the indexed performance measure and sustainability. Such assumptions are unevenly justified.

A model guiding restoration to a fully natural biodiversity based on existing reference conditions, including some idea of the abundance of significant resources, involves the least risk that resources of significance will fail to be restored as forecast as long as the significant resources are also found in the natural reference conditions. They are also most likely to restore a sustainable state, if the existing natural state is sustainable. However, the influential landscape variables that often determine local sustainability are frequently not addressed in most existing models, only a few process simulation models approach this level of comprehensiveness (e.g., DeAngelis et al. 1998). For example, major changes in precipitation, air temperature, cloud cover, and sea level could greatly modify, even eliminate many existing ecosystems over a period of 50 years. Models incorporating more than one measure (multicriteria models) of biodiversity/ integrity are more likely to inclusively represent natural biodiversity than most single-species models. While single species models can be useful when chosen with the entire ecosystem in mind, or (sometimes) as indicators of resource significance, they are easily misused.

Many restoration proposals target partial restoration of naturalness under more complicated conditions involving much fragmentation of the original ecosystem conditions and many different sources of stress and pathways to altered states. They involve systems with many natural and human influences, interactive feedbacks, landscape-scale considerations, remotely located and subliminal limiting factors, and other complex interactions, such as occur in many culturally modified parts of ecosystems. As conditions grow more complicated, the advantages of spatially explicit process simulation models begin to outweigh the accessibility and low-cost advantages of other models. Regardless of model choice, when partial restoration of ecosystems is under consideration, the relationship of output indicators for resources of significance and output indicators of naturalness need to be defined clearly to assure consistency with Corps restoration policy.

4.4.2 Modeling For Common and Scarce Biodiversity

The previous review of ecological principles in Section 3 suggests that some multi-dimensional measure of natural biodiversity may hold promise as an indicator for most, if not all, of the non-monetary benefit sustained by fully natural ecosystems. Several types of "biodiversity" models characterize relationships between habitat inputs and community or ecosystem outputs along a gradient of human effect anchored in the most natural condition. However habitat and community measures of biodiversity in most existing model types are most reliable for the more common ecosystems components and aggregate function and structure. They often loose predictive reliability for the scarcest components, most likely to qualify as resources of significance, such as the globally rare species, as indicated in Figure 3.5 and Figure 4.1. This deficiency has to do with the practical problems associated with calibrating models, which typically are based on the more common components of ecosystems.

Even species diversity measures frequently miss explicit inclusion of the globally rare species, which often qualify as the resources of greatest national significance. Thus the habitat-community relationships defined for the more common species must be assumed to hold for the rarest species as well. This assumption becomes increasingly secure as more the ecosystem needs of more of the species the community are included in the model. Even so, the uncertainty associated with inherent variation, often determined by random events, increases as the restoration justification increasingly hinges on the response of a very few species to restoration measures. Few commonly used models have addressed this uncertainty issue adequately.

When the resources of significance are based on scarce biodiversity, as indicated by the uniqueness and vulnerability of species, model selection depends on how many of the species in a community qualify as significant resources. In a situation where only one or two species qualify as scarce in a restored community, it would typically be best to use a community model to guide the restoration of the natural support system, and species-habitat models to check on whether or not the suitability of habitat has been obtained for the significant resource species. As more species in a community qualify for scarcity status, the added benefit of coupling with individual species-habitat models decreases. For greatest utility, the coupling of community and species models requires that all of the input conditions (habitat variables) for the species-habitat model also would be included in the community-habitat model. This will require a coordinated effort that has yet to be done. Thus, the most useful models for recovering overly scarce biodiversity have yet to be developed, either in index form, the more elaborate form of process simulation models, or in hybrid models.

Existing methods and models can be usefully applied to formulate and evaluate for scarce biodiversity resources, but with heavy reliance on professional judgment and concept models of the system context. Concept models should be developed with special attention to the risks and their management. Once species, guilds or entire communities have been determined to qualify as resources of significance, the primary challenge is to identify the risk of project failure in realizing their recovery and managing that risk to an acceptable level. As a general rule, risks are lower when the project area to be restored is immediately adjacent to and functionally closely tied to a large, fully natural area that supports thriving remnants of scarce resources, and when the causes of degradation are few and easily corrected. As the project area becomes more disconnected from the naturally intact ecosystem and the causes of degradation become more numerous and complex the risk of realizing a sustainable contribution to NER increases.

The existing set of modeling tools are more reliable for restoration plan forecasting when the resources of significance are determined to be associated with restoration of the more common biodiversity in ecosystems—such as the production and biomass functions that contribute substantially to aesthetic, recreational, flood damage reduction, water supply, and water quality services. The tools are more dependable because the resources are abundant enough to have been well studied, in contrast with the scarce resources. However, if resource scarcity is the most important determinant of NER qualification, substantial improvement of existing models and methods is in order.

4.4.3 Existing Model Limitations

Few existing models can be used without extreme care and understanding of the underlying project ecosystem condition and its systems context. While species-based HSI models are numerous, easy to use, and immediately available, and are relatively inexpensive (Figure4.2), they rarely capture all of the important habitat/ ecosystem elements to assure a more natural, self-regulating condition will result, or all of the justifying value needed to restore ecosystems. Species-based HSIs are not scaled based on ecosystem integrity and can only be used to indicate a more naturally integrated ecosystem condition if the HSI value is known for the targeted restored condition. Few existing single-species HSI models satisfy these criteria well, but ecosystems might be characterized by new models for native dominant and keystone species, including dominant plant species, scaled against a gradient of altered conditions anchored in the most natural ecosystems. Several species HSIs might be used to "bracket" the community-habitat relationships satisfactorily, but the need for many new models and much calibration offsets the main existing advantage of HSI models. In addition, few HSI models now exist for the most vulnerable species or guilds in aquatic ecosystems, and would need development for use either alone or with models of ecosystem naturalness.

Community HSIs indicate relative ecosystem naturalness and associated non-monetary benefit more inclusively than species-based models because they link habitat more broadly to ecosystem components or functions. Among existing models, WVA appears to have many of the same limitations of the species HSIs from which it was derived. It is based on the optimum needs of relatively common species; not on a scale of relative naturalness or on scarce resources of environmental significance. The HGM approach l links directly to the naturalness of ecosystem functions through FCIs, but, like all index models, they cannot be readily compared across local ecosystem conditions to aid in restoration priority decisions.

Sustainability of ecosystem function and structure is an increasingly important criterion for model selection, and the closely related concept of self-regulation is a defining attribute of more natural conditions in Corps policy. However, concepts of ecosystem health and cultural integrity suggest that sustainable states can coexist with substantial human alteration in carefully considered situations. Principles of forest, range, and other natural resource management have assumed such for many decades, sustainable management being a cornerstone of wise resource use. Index models do not address functional stability, self-regulation, and sustainability of ecosystem structure explicitly, however. Species HSI models usually provide little theoretical or practical insight into the sustainability of the conditions they indicate. While it might be assumed that the FCIs of HGM, or the HSIs of communities are proportional to an ecosystem's capacity for self-regulation, functional stability, and sustainability of structure, these attributes of ecosystems have not been examined critically. Because all of these models focus on local conditions, they fail to capture all of the landscape attributes of the entire ecosystem that are so important in determining sustainability of scarce ecosystem structure.

Models vary in the extent to which they have been developed. By far the greatest number if models available "on the shelf" are single-species HEP/HSI models. But few existing models appear suitable for environmental resource evaluation. The ecosystem index models that have the greatest potential for use in a wide variety of ecosystem types are the IBI, FCI of HGM, and WCHE, but none have been developed for a full range of ecosystem conditions of interest to the Corps. The IBI has the longest history and diversity of development, but even among stream ecosystems for which it is best developed, many stream ecosystems remain to be calibrated. HGM has yet to cover all wetlands let alone all other ecosystems of interest to the Corps. The WCHE is most limited in this regard, having been developed only for one type of forested wetland and several upland ecosystems. Any of the ecosystem index models would require considerable investment to cover the variety of ecosystems managed by the Corps, but IBI and HGM have had the greatest investment so far. Integration of IBI, HGM, WCHE and other index model attributes is a possibility that ought to be considered as well, if an index approach is to be emphasized in the future.

Ecosystem index models also make broad assumptions about the "tightness" of relationship between selected indicator species and the entire ecosystem. Most models disproportionately rely on fish, invertebrate, or bird subsets of the community-habitat relationship to represent the entire ecosystem condition. The taxonomic groups chosen for characterizing integrity may not characterize to fine enough degree all of the relevant attributes of a more natural condition, nor the habitat needs of the scarce resources of significance. Complete methods would need to account for this potential deficiency by assuring the biodiversity measure in the index is inclusive of the significant resources or by including a separate relationship between vulnerable-species and habitat conditions.

Many of the shortcomings of index models are addressed in process simulation models, which ultimately offer the greatest flexibility in use and the greatest management insight with respect to the output generated with incremental additions of restoration measures. Self-regulating mechanisms are built into such models through density dependent and other feedback relationships. Functional stability and sustainability can be analyzed directly from the dynamics of modeled output, but still remain among the more difficult attributes to model. Functional and structural changes can be examined in explicit estimates of actual output amounts and in spatially explicit dimensions. The effects of uncertainty can be assessed through analysis of the sensitivity of output to the uncertainty associated with specific model structure. Process simulation models are more typically "theoretically rigorous" because process understanding is an important objective. Because of this, they are among the best models for organizing information adaptively through time as new information becomes available. In terms of basic ecosystem structure, processes, and interactions, similar principles operate across all ecosystems to which such models apply.

However, process models can be "information hungry" and more time consuming, especially when precise prediction is a high priority. Much can be learned about how ecosystems work during assembly of process models, but the ultimate models for evaluating nonmonetized environmental benefits are years away even if research investment were immediately and substantially increased . The objections to process models expressed two decades ago (leading to the emphasis on index models), having to do with inadequate portability and computational capability, have been greatly diminished by the widespread availability of powerful personal computers. Even so, the details of resource partitioning into communities of different species richness and functional stability requires much research and development. In the process of assembling such models, much more could be learned than from index models about managing ecosystem process for more reliable service delivery across all natural and enhanced services. Process simulation shows the most promise for incorporating tradeoff analysis within single model operations.

The existing state of ecological knowledge, management need, and model development capability leads to the conclusion that, in the near term, a selection of environmental-benefits estimation models needs to be made available for resource management planners. If recovery of resources of significance and ecosystem naturalness are to be jointly considered objectives of ecosystem restoration, the most useful planning models will be capable of representing the responses of both to restoration measures in naturally variable settings. In the short-term, this may require a combination of a suitable community HSIs or ecosystem FCIs and species or guild HSIs. Much development is needed, however, because many ecosystems and resources are not now addressed by existing models. In the longer term, greater development and use of process models ought to be considered because of their more explicit estimation of actual output amounts, their capacity for organizing great amounts of model input and process information into simultaneous forecasts of numerous and diverse outputs, and their long term adaptability to management needs.